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Research article

Is wind energy increasing the impact of socio-ecological change on Mediterranean mountain ecosystems? Insights from a modelling study relating wind power boost options with a declining species

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ABSTRACT

The growing concern about future challenges of energy security and climate change has led to the expansion of renewable energy production, with a special emphasis on wind power. Despite the environmental advantages of wind power, it's important to assess the impacts caused by the presence of wind farms on wildlife, particularly on species also affected by habitat loss and degradation. In Mediterranean Europe, the skylark (*Alauda arvensis*) is a declining passerine that breeds in mountain habitats vulnerable to the abandonment of traditional management practices and climate change. We have created a spatially explicit agent-based model (ABM) in order to replicate the selection of territories, evaluating the effect of wind farms on the mortality rate of breeding males. We were especially interested in assessing the mortality rates related with the interplay between habitat loss due to socio-ecological change and increasing wind power using alternative strategies: adding wind turbines or substituting existing wind turbines by more powerful ones, i.e. repowering. Several known aspects related with the risk of collision of *A. arvensis* with wind turbines were considered, particularly regarding the male habitat selection and behaviour displayed throughout the breeding season. By simulating a sequential contraction of suitable habitat for the species, we found a substantial increase in the breeding territories superimposed to the wind farm influence zone. In these conditions males' relative mortality was predicted to suffer significant increases. For equivalent wind power, adding wind turbines produced significant increases in the males' relative mortality, whereas repowering didn't. Based on our findings we propose repowering as a defensible strategy to increase wind energy production without increasing *A. arvensis* collision risk. We highlight that this strategy might also benefit other vulnerable bird and bat species associated with declining habitats of mountain ridges in the Mediterranean region.

1. Introduction

Climate change and the worldwide increasing energy demands have been acknowledged as the greatest environmental challenges of the 21st century (Zhang, 2008; Rey-Hernández et al., 2018). Novel climates, exhibiting conditions with no analogue in the observational record, interact with local anthropogenic environmental degradation (socio-ecological change) leading to habitat loss and fragmentation with serious consequences on biodiversity and ecosystems integrity (Thomas et al., 2004; Hoffmann et al., 2010). Species' responses to the effects of socio-ecological change include shifts in range as well as behavioural adaptation (Bellard et al., 2012). The actual rates of habitat loss and fragmentation might hinder or impede species' responses, accentuating declines in populations by reducing genetic diversity, dispersal ability and reproductive success (Fahrig, 2003; Fischer and Lindenmayer, 2007; Segan et al., 2016). Additionally, increases in the exposure to invasive species and vulnerability to stochastic events alters species relations and ultimately might originate local extinctions (Fahrig, 2003; Fischer and Lindenmayer, 2007; Segan et al., 2016).

In order to reduce climate change, countries worldwide have been

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committed on implementing sustainable energy strategies, focusing on more efficient technologies and alternative energy sources (Saidur et al., 2011; Barreto, 2018). Wind power plays a significant role in current and future energy demands, corroborated by public policies towards reducing greenhouse gases emissions and private investments in this sector (Silva et al., 2006; Forte et al., 2010; Sims et al., 2011; Rogelj et al., 2016; Raimundo et al., 2018). Despite the global environmental benefits of wind power, the determination of suitable wind farm locations requires not only an evaluation of the potential profitability, but also a comprehensive assessment of the possible impacts resulting from its installation (Langston and Pullan, 2003; May et al., 2015; Bastos et al., 2016; Ferreira et al., 2015; Silva et al., 2017). Environmental impacts must be considered, including the effects on flying vertebrates, such as birds and bats (Percival, 2000, 2005; Drewitt and Langston, 2006; Larsen and Guillemette, 2007; Pearce-Higgins et al., 2009; Eichhorn et al., 2012; Santos et al., 2017).

Wind farms are generally located on mountain ridges in the Mediterranean region, overlapping threatened habitats which are important for several vertebrates as foraging grounds, migration routes and nesting areas (Orloff and Flannery, 1992; BirdLife International, 2003; Travassos et al., 2005; Santos et al., 2010). Birds represent one of the most affected groups by the presence of wind farms, primarily due to collision fatalities and habitat loss (Drewitt and Langston, 2006; Santos et al., 2010; Pearce-Higgins et al., 2012; Marques et al., 2014; Schuster et al., 2015; Smith and Dwyer, 2016). Most fatalities by collision occur during foraging periods, daily movements towards the overnight sites or during larger scale migration movements (Smith and Dwyer, 2016). Specific flight patterns performed by several birds (and bats) are considered particularly perilous (Berkhuizen and Postma, 1991; Morinha et al., 2014; Bastos et al., 2016; Thaxter et al., 2017) and weather conditions appear to influence substantially the risk of collision, i.e. birds tend to fly at low altitudes, facilitating collisions with wind turbines on days with overcast conditions, fog, rain, strong winds and dark nights (Barrios and Rodriguez, 2004; Kingsley and Whittam, 2005; Travassos et al., 2005; Madders and Whitfield, 2006; Morinha et al., 2014; Aschwanden et al., 2018). Additionally, the location of the wind turbine seems to play an important role on collision related bird fatalities (Berkhuizen and Postma, 1991; Erickson et al., 2001; Morrison, 2006; Thaxter et al., 2017). Turbine design and characteristics are also likely to influence collision risk (Thelander et al., 2003; Winkelman, 1992).

Although formerly considered a common species and evaluated as “least concern” (BirdLife International, 2018), the Skylark, *Alauda arvensis* (Linnaeus 1758), has suffered a large decline since the 1970s in all Western European countries (BirdLife International, 2004; Burfield and van Bommel, 2004; Julliard, 2005; IUCN, 2012). Conservation efforts have been made to halt and reverse the declining trends observed in the last decades (Copland et al., 2012). Taking into account the vulnerability to predatory phenomena, skylarks tend to avoid areas near forests, hedges, isolated trees, shrubs and other vertical structures, nesting preferably in the ground (Donald et al., 2001; Daunicht et al., 1996). In the Mediterranean region the species occurs in a variety of open habitats, albeit restricted in the breeding season to cool and wet mountain pastures and dwarf shrublands (High conservation interest and environmental sensitivity Annex I habitats, EU Habitats Directive - http://natura2000.eea.europa.eu/) associated with traditional animal husbandry based on transhumance (e.g. Coelho, 2007; Chamberlain and Gregory, 1999; Browne et al., 2000; Donald, 2004; Tucker and Heath, 1994).

Depopulation, abandonment of agriculture, decreases in the traditional animal husbandry and climate change (socio-ecological change) have been shifting the Mediterranean mountain habitats and species, namely by substituting wet pastures and dwarf shrublands by dense shrublands and forests (e.g. Lasanta-Martınez et al., 2005; Rumpf et al., 2018; Sanjuán et al., 2018). Climatic-induced contractions and upslope range-shifts in mountain birds, possibly as a consequence of adverse impacts of higher temperatures on physiology, competitive interactions and nest predation were already noticed for *A. arvensis* (and co-occurring bird species) with a rise of 50 m per decade (e.g. Flousek et al., 2015; Lehikoinen et al., 2018). Negative trends in pluviometry seem to be creating similar shifts in mountain plant communities, namely by substituting wet grasslands by more xerophytic shrulands (García-Romero et al., 2010; Tietjen et al., 2017). In addition, Mediterranean mountains have now a reduced livestock pressure that has resulted in enhanced natural secondary succession: field studies have documented the process of vegetation succession from the valleys to the ridges, and confirmed that within a time frame of 10 years ungrazed mountain pastures were completely covered by woody vegetation (e.g. Sanjuán et al., 2018; García-Llamas et al., 2018).

During the breeding season, *A. arvensis* males are fiercely territorial, defending aggressively females and the area immediately around the nest. To fend off rival males and birds of other species, skylark males produce long and continuous song flights with a circular trajectory (Catchpole and Slater, 1995; Donald, 2004; Briefer et al., 2010). The aerial exhibition of the song flight and territorial behaviour increase the risk of mortality due to collision with rotating blades of wind turbines. Actually, *A. arvensis* is one of the species with higher mortality rates due to collision with wind turbines in European wind farms (e.g. Hötker et al., 2006; del Rio and Fontán, 2006; Coelho, 2007; Profico Ambiente, 2002; Bernardino et al., 2010; Zieliński et al., 2011; Bernardino et al., 2012; Rydell et al., 2012; Morinha et al., 2014; Schuster et al., 2015; Bastos et al., 2016; Zehtindjiev and Whitfield, 2016). Furthermore, the study conducted by Morinha et al. (2014) has proven a sex and age biased mortality (affecting mostly adult males), relevant findings for outlining mitigation and conservation actions for the species (Ewen et al., 2001; Buenestado et al., 2009).

Considering the declining trends and the substantial amount of fatalities of *A. arvensis* recorded in wind farms, it is crucial to carry out robust studies to understand the impacts of socio-ecological changes in the mortality and viability of populations, in order to establish suitable measures of mitigation and conservation (Beston et al., 2016). Bastos et al. (2016) developed an integrative spatially-explicit dynamic approach to predict the actual and future cumulative regional impacts of wind farms on Skylark breeding populations. Nevertheless, local impacts associated with specific wind power projects must be gauged using detailed wind farm information linked with the species behavioural characteristics (e.g., Dupont et al., 2016). A promising possibility to investigate the interplay of species behaviour in specific landscapes involves the development and implementation of spatiotemporal models using agent-based modelling platforms (ABM) (DeAngelis and Mooij, 2005; Grimm and Railsback, 2005; Eichhorn et al., 2012; Ferreira et al., 2015). ABMs are computational models that use the basic constituents of a system (i.e., individual agents), to reveal the emergence of system properties from the interactions between its various components (Grimm, 1999; Grimm and Railsback, 2005). Several features can distinguish ABMs from other more traditional modelling approaches, such as spatiotemporal variability and the ability to include individual behaviour allow emerging population responses to both environmental variables and species characteristics (Dalkvist, 2011; Ferreira et al., 2015; Manore et al., 2015; Santos et al., 2016, 2017).

This paper outlines the development and application of a spatiotemporal explicit ABM model to forecast local Skylark mortality risk under the influence of changing landscapes and wind power development options. Specifically our spatiotemporal explicit model should: (1) predict territory settlement and densities for changing landscapes, providing relevant information of the estimated breeding population of *A. arvensis*; (2) simulate, for each territory, the male song flight aerial demonstrations; (3) evaluate the risk of mortality, assessing the probable impact associated with the interplay between changes in the area of suitable habitat and strategies for increasing energy power (4). contribute to increase our knowledge on the interaction between

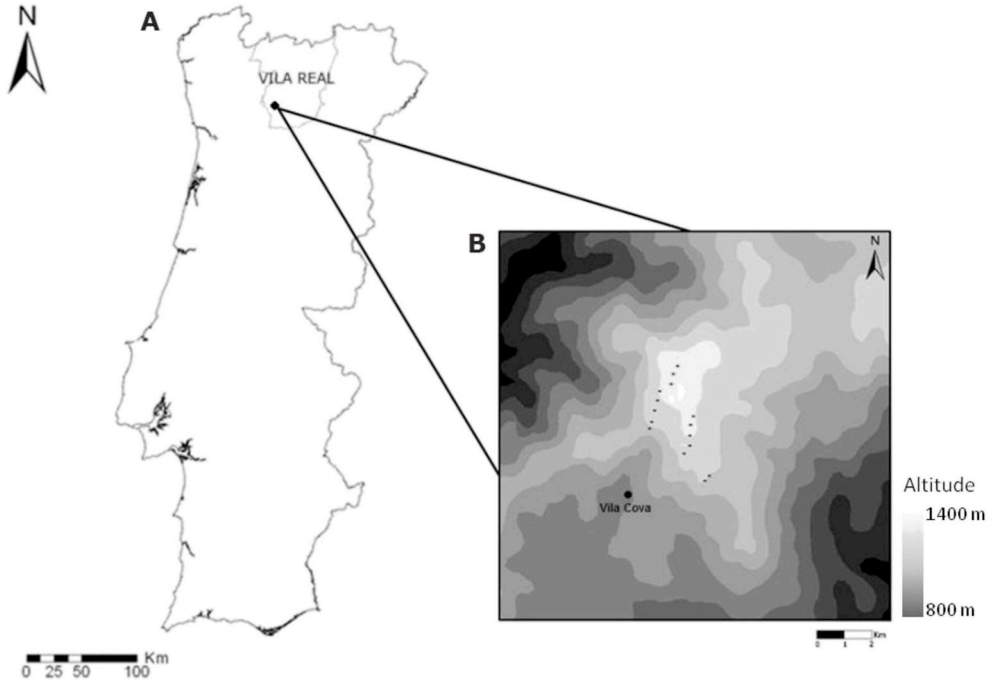

Fig. 1. Location of the study area in the district of Vila Real, north Portugal (A), and a detail of the Outeiro wind farm where the circles indicate the position of the wind turbines (B).

declining species, landscapes under socio-ecological change and wind energy exploitation, contributing for mitigation strategies and decision-making to overcome possible conflicts.

2. Methods

2.1. Study site description

The model was developed for a 10,000 ha landscape centred in the Outeiro wind farm, located in northern Portugal (Fig. 1).

The wind farm consists of 15 wind turbines (64 m towers, 70 m rotation diameter), arranged along three ridge lines at altitudes ranging from 1186 m to 1311 m. This area presents an altitudinal gradient between 800 and 1311 m. Low visibility conditions are common and occur in approximately 66% of the year, particularly in the form of fog, moderate cloudiness or overcast (Profico Ambiente, 2002; LEA, 2010). The vegetation is dominated by high priority Natura 2000 wet grasslands and dwarf shrublands (Annex I of habitats directive; Mãe d'Água, 2006). These habitats constitute important nesting and feeding grounds for several species of mountain birds, including *A. arvensis*, *Prunella modularis*, *Emberiza cia*, *Saxicola rubicola*, *Sylvia undata*, *Monticola saxatilis, Anthus campestris* and *Circus pygargus* (Mãe d'Água, 2006). If we consider a more comprehensive area encompassing the Alvão/Marão Natura 2000 site (PTCON003), biotopes of high conservation concern can be found with a significant diversity of endemic/restricted distribution species (ICN, 2005; http://natura2000.eea.europa.eu/#).

2.2. Modelling procedure

The conceptual description of the model follows the specifications of the standard protocol ODD (Overview, Design Concepts and Details) to describe ABMs, proposed by Grimm et al. (2006) and reviewed in 2010 (Grimm et al., 2010). The Overview outlines the overall purpose and structure of the model (Grimm et al., 2006). Design Concepts describes the general notions underlying the design of the model integrating this approach into the larger framework of Complex Adaptive Systems (Grimm and Railsback, 2005). The Details section presents specificities omitted in the Overview providing relevant information to re-implement the model (Grimm et al., 2006). All simulations were performed using NetLogo 6.0.4. software (Wilensky, 1999).

2.2.1. Overview

2.2.1.1. Purpose. The ABM main purpose was assessing trends in Skylark mortality risk caused by the interaction between the reduction in the area of suitable habitat, attributable to ongoing socio-ecological changes - climate induced uphill shifts in wet meadows, dwarf shrublands and Skylark populations plus significant vegetation modifications associated with animal grazing reduction - and different strategies to increase wind power. Using this context as a demonstration we were especially interested in understanding the best strategy for increasing wind power while mitigating the impacts on vulnerable species.

2.2.1.2. Entities, state variables and scales. The model features three entities, briefly described in Table 1, the patches (unitary landscape parcels), male skylarks and wind turbines. The model simulates a 10,000 ha square (corresponding to 295,936 patches), centred on the Outeiro wind farm. The virtual world is closed and does not allow agents entry or exit. Skylarks are the only mobile entities, which are programmed to reproduce the specific breeding season behaviour of male *A. arvensis*. From this point forward, virtual individuals will be referred to as *skylark(s)*, while real birds will be referred by the species scientific designation *A. arvensis*. The wind turbines have fixed coordinates that simulate their real location within the study area.

The model is divided into two distinct stages, the first simulates *A. arvensis* territory establishment processes (Territory Establishment Stage – TES), while the second recreates the typical territorial behaviour displayed throughout the breeding season (Breeding Season Stage – BSS). The second stage of the simulations includes mortality risk related to wind turbine collision. The TES simulations unfold with no correspondence between the tick (default discrete time unit) and any real time unit, as this phase ends only when all areas of suitable habitat for territory establishment are occupied. Additionally, we did not intend to simulate the actual amount of time spent by *A. arvensis* during

Table 1
Description of the variables of each conceptual entity used in the ABM model.

Entity	Variable	Brief description
Patches	*Number*	- 295,936 patches arranged in a square grid (10,000 × 10,000 m), which corresponds to a total of 10,000 ha.
	Size	- Individual patch – 0.034 ha square (18.38 × 18.38 m).
	Altitude	- Altitude is set in 100 m intervals. All patches that lie between two altitude lines show the value of the lower curve.
	Land_cover	- Each patch contains a single class of land cover (class 1 - land cover with favourable conditions for territory establishment; class 2 - land cover with unfavourable conditions for territory establishment).
	Buffer	- Area around the wind turbines considered in the Breeding Season Stage.
	Risk	- Risk value for the patches located in the influence zone of the wind turbines. Depends on visibility conditions.
	Free	- Reports if the patch is useable for territory establishment: class 1 land cover, altitude requirements fulfilled, no territory already occupied by another male.
Skylarks	*Number*	- Number of individuals per simulation.
	Size	- 0.018 *patches* based on average 33 cm wingspan according to Cramp (1988).
	Potential_radius	Radius of the potential territory initially assigned randomly to each individual according to the intervals tested in the Territory Establishment Stage simulations.
	Territory?	- Distinguishes individuals that have established their territory (*Territory?* = 1) from those who haven't (*Territory?* = 0).
	Time_count	- Time count which determines the timing and duration of the song flight.
	Height	- Represents the height reached from the ground.
	Out_of_territorry	- Time counter activated when individuals leave their territory. Sets the maximum amount of time spent outside the territory.
	Avoid	- Random value (between 0 and 99) assigned by tick, that determines a mortality event if the *skylark* is in a risk patch.
	Flight_count	- Records the number of successful song flights executed in each day.
	Flight_frequency	- Defines the initial periodicity of the song flights for each month.
	Flight_duration	- Counts the total amount of *ticks* spent during the song flights.
	Territory_size	- Reports the number of patches of the territory.
	Song_flight	- Total duration of the song flight (*upward_flight* + *horizontal_flight* + *downward_flight*).
	Upward_flight	- Duration of the ascending stage of the song flight.
	Horizontal_flight	- Duration of the horizontal flight stage of the song flight.
	Downward_flight	- Duration of the descending stage of the song flight.
	Flight_angle	- Randomly assigned angle (between 15 and 30°) that defines the amplitude of the circular trajectory of the horizontal stage of the song flights.
	Total_height	- Sum of all heights achieved during the song flights.
Wind turbines	*Number*	- 15 wind turbines or 20 wind turbines.
	Size	- 3.8 patches or 5.4 patches (rotor diameter).
	Coordinates	- Coordinates of the fixed position of the wind turbine in the virtual world.
	Orientation	- Daily random orientation (between 0 and 359°).

this process. On the other hand, the time unit used during the BSS simulations was the second as it is the discrete unit that better adjusts to *A. arvensis* movements, particularly during the song flights when collision risk is critical. This stage simulates the five months breeding period (March, April, May, June and July) which starts immediately after the territory establishment. Taking into account the extent of the study area as well as the time length of the second stage of the simulations, we focused BSS simulations in a smaller area in the immediate vicinity of the wind turbines: wind turbine influence zone. The wind turbine influence zone was estimated for each rotation diameter model given that *A. arvensis* territories do not tend to exceed 5 ha in northern Portugal, with an average of circa 4 ha (LEA, 2011; LEA, 2012; Mãe d'Água, 2006). We considered that individuals with territories in outlying areas of the wind turbine influence zone had very low probability of collision with these structures, once the song flights are performed almost exclusively within the limits of the territory (Donald, 2004).

2.2.1.3. Process overview and scheduling. The simulations followed a sequence of procedures in order to mimic the typical behavioural stages shown by *A. arvensis* males during the breeding season, from March to July, as described by several authors (e.g. Lange, 1951; Delius, 1965; Catry et al., 2010). All procedures described below are briefly presented by the flowchart for the TES (Appendix A) and the BSS (Appendix B) in the Supplementary Materials.

2.2.1.3.1. TES simulations

2.2.1.3.1.1. Individual distribution. Each simulation starts with the distribution of the *skylarks* over their initial positions in the virtual world. Each *skylark* is placed randomly on patches with favourable conditions for territory establishment according to land cover and altitude parameters: area of suitable habitat. This procedure simulates the arrival of individuals from the wintering areas, once *A. arvensis* males show a tendency to return to the same territory or nearby areas, demonstrating a strong fidelity to territory sites (Delius, 1965). The habitats that compose the virtual world were divided in two classes according to the specific characteristics of the vegetation (Appendix C). Class 1 consists of a series of habitats considered appropriate for territory establishment by presenting short vegetation height, used by *A. arvensis* for foraging (Donald et al., 2001; Toepfer and Stubbe, 2001). This criterion was further reinforced by the requirements for the nesting site, generally occurring in areas with less than 60 cm vegetation height (Daunicht et al., 1996). All habitats with low quality in terms of foraging and nesting potential were grouped into Class 2.

Mountain habitats appear to play a key role during the breeding season for this species in Mediterranean Europe (Tucker and Heath, 1994; Browne et al., 2000; Chamberlain and Gregory, 1999). In north Portugal wet pastures and dwarf shrublands above 800 m are the most important habitats used during the breeding season, although higher thresholds may be chosen: in Spain, usually under drier conditions and where abandonment of agriculture and mountain pastures is more pronounced, the species breeds mostly at altitudes above 1000 m (Pimenta and Santarém, 1996; Suárez et al., 2003; Catry et al., 2010; Equipa Atlas, 2008). The minimum altitude allowed for territory establishment in the model can be set manually in order to evaluate the effects socio-ecological change in the area of suitable habitat, i.e. increasing the altitude where habitats and environmental conditions suitable for the species reproduction start occurring (Bastos et al., 2016). Multifactorial effects of socio-ecological change in mountains are mostly associated with an upward range shift (Peñuelas and Boada, 2003; Sanz-Elorza et al., 2003; Lenoir et al., 2008; Pauli et al., 2007; Vittoz et al., 2008; Harris et al., 2014; Avalos and Hernández, 2015; Bastos et al., 2016; Virkkala et al., 2018; Rumpf et al., 2018). In this way, three hypothetical classes of minimum altitude for the species occurrence (area of suitable habitat), 800 m, 1000 m and 1200 m were simulated. In each simulation individuals were only distributed in patches with altitude equal or superior to the set-out minimum. This procedure aimed to assess how a sequential reduction of area of suitable

habitat (ASH), mostly associated with upslope vegetation succession due to grazing abandonment and climate-induced shifts, may interfere with the territory selection and establishment dynamics.

2.2.1.3.1.2. Potential territory size definition. In the initial distribution of *skylarks*, each individual is assigned a preliminary radius value that will define the area of the potential territory (*Potential_radius*). These values were assigned randomly within limits obtained in the Wind Farm Monitoring Report that pointed to territories ranging from 3 to 5 ha (Mãe d'Água, 2006).

2.2.1.3.1.3. Territory establishment. The territory establishment procedures are mainly based on the assumption that *A. arvensis* males only select and defend territories in areas where minimum quality requirements are met, particularly in terms of habitat. Before executing exploratory movements in search of territory, each individual will check if their current *patch* is free from other male occupation and offers ideal conditions of altitude and habitat (area of suitable habitat) by assessing the *Free* variable. If the *patch* does not provide the required conditions (*Free* ≠ 0) the territory settlement is made unviable and the individual will enter the territory search mode (see paragraph 2.2.1.3.1.5.). On the other hand, if their positional *patch* meets all the requirements (*Free* = 0), *skylarks* will examine the *Free* variable of all *patches* comprised in the surrounding area defined by the preliminary radius value previously assigned (see paragraph 2.2.1.3.1.2.). When the area resulting from the sum of all good quality patches is inferior to the minimum territory size initially set for the simulation, the potential territory size enlargement procedure is initiated (see paragraph 2.2.1.3.1.4.). However, if the area resulting from the sum of all good quality patches is equal or greater than the minimum territory size initially set, the procedures for territory establishment are triggered: The *skylark* interrupts the searching movement; the *Free* variable of all patches comprised in the area defined by the radius value changes from 0 to the identification number of the *skylark*, thus creating a link between individuals and all patch of their territory; the *Territory?* variable of the *skylark* goes from 0 to 1 and the territory size is registered by the *Territory_size* variable for further analysis.

2.2.1.3.1.4. Potential territory size enlargement. This procedure is only applied when a roaming individual is positioned in a *patch* that meets the habitat and altitude requirements (area of suitable habitat) and simultaneously hasn't been occupied by another male. If these conditions are met and the area defined from the *Potential_radius* variable does not encompass enough land parcels to ensure a quality territory, *skylarks* will enlarge their potential territory size by adding 1 unit per time unit to the *Potential_radius*. This procedure will enable the definition of larger potential territories in order to ensure that a minimum of quality *patches* is attained for territory establishment. Individuals in territory size enlargement mode will remain in their current position until the territory is implemented or the maximum territory size value for the current simulation is reached, in which case they will enter the territory search mode (see paragraph 2.2.1.3.1.5.).

2.2.1.3.1.5. Territory search. The territory search procedure is initiated when the minimum conditions for territory establishment are not met. Territory seeking Individuals move randomly within a 10 patch radius of their current position for every time unit. After each movement all conditions for territory establishment are checked again in order to assess if the territory search mode must continue.

2.2.1.3.1.6. Finalization. TES simulations stopped when all patches with territory potential are occupied. The requirements for the simulation finalization are assessed once every time unit.

2.2.1.3.2. BSS simulations

2.2.1.3.1.1. Time structure and global variables. Each simulation represents the breeding season, corresponding to a five months period, given that in Portugal breeding individuals can be heard defending their territories from March to July (Catry et al., 2010). The time unit used for the BSS simulations was the second, considered appropriate to simulate *A. arvensis* flying movements (Hedenström, 1995), detect mortality events and possible deviations in the modelled behaviour from the expected behaviour. We also divided the simulation in time periods that correspond to both days and months. According to Keyser (1999), during the breeding season *A. arvensis* males start singing right before sunrise and remain active throughout the day decreasing their activity at sunset, and ceasing activity completely during the night. The day length was based on data from the Lisbon Astronomical Observatory (Supplementary Materials: Appendix D). At the end of each day all relevant data was recorded, one day unit was added to the day counter and the time count restarted. At the end of the month, an automatic transition to the next month occurred updating the day duration and monthly number of days. The corresponding adjustment of the variables *Flight_frequency* (defines the initial periodicity of the song flights for each month) and *Song_flight* (defines the typical duration of the song flights) also took place at this stage. The simulations stopped at the end of July.

Several authors highlight wind speed and visibility as some of the most relevant weather variables with influence on collision risk (e.g. Travassos et al., 2005; Skov and Heinänen, 2015; Marques et al., 2014). The weather conditions were assigned randomly from pre-established classes on the beginning of each simulated day. *A. arvensis* males perform short upwind movements on windy days (> 6 m/s), and execute more complex song flights on days with wind speeds equal or below 6 m/s. So, on days with lower wind speed, the "aerial territory" is larger once they show a characteristic circular flight trajectory (Donald, 2004). The wind speed variable was divided into two classes: class 1 – wind speed ≤6 m/s and class 2 – wind speed > 6 m/s. Although several authors refer to an increased risk of bird collision with wind turbines in low visibility conditions (e.g. Travassos et al., 2005; Longcore et al., 2013; Marques et al., 2014), there is still a lack of accurate data to allow a precise quantification of the influence of visibility in the risk of collision (see paragraph 2.2.1.3.1.5. Mortality). Visibility conditions were divided into three classes (class 1 – bad visibility; class 2 – medium visibility; class 3 – good visibility).

2.2.1.3.1.2. Initial male population configuration. BSS simulations feature male mortality events and the consequent occupation of their territory by other roaming individuals. According to Donald (2004), *A. arvensis* individuals born in the previous year exhibit strong nesting site fidelity, showing a tendency to return to an area of about 1 km around the nesting site where they were born. This confluence of individuals implies that some of the males are unable to establish their own territory eventually integrating a non-breeding population (Donald, 2004). According to Delius (1965), these individuals represent about 10% of the total population in breeding areas. Thus, the initial male population definition procedure includes the removal of 90% of individuals without territory, whilst the remaining 10% of these individuals remain active forming a roaming population subject to the same procedures implemented in TES simulations.

2.2.1.3.1.3. Movement on the ground and song flights. The procedures that determine individual movement on the ground depend on the *Territory?* variable which indicates whether or not the *skylark* has established territory. Thus, individuals that belong to the roaming population (*Territory?* = 0) keep executing the procedures employed in the TES simulations. These individuals will reoccupy the free areas that result from the death of other males due to collision events (Penteriani et al., 2011).

On the other hand, all individuals that were able to establish their territory in the TES simulate the typical behaviour presented throughout the breeding season. During this period, *A. arvensis* males are fiercely territorial, defending the respective females and the area immediately around the nest aggressively and generally performing most of their song flights over this area. To fend of rivals, males produce long and continuous song flights, unlike many other birds, which normally produce relatively short and discontinuous song flights (Catchpole and Slater, 1995; Donald, 2004; Briefer et al., 2010).

The typical duration of individual song flights (*Song_flight*) for each month was recreated based on the study conducted by Lange (1951),

Table 2
Average number of *Alauda arvensis* male song flights per day; individual song flight duration; daily song flight duration and song flight frequency based on Lange (1951).

Month	Song flights per day	Individual song flight duration (seconds)	Daily song flight duration (seconds)	Flight frequency (seconds)
March	4.45	120	534	10,370
April	10.13	180	1854	4835
May	11.26	186	2094	4791
June	21.3	270	5751	2465
July	45.74	330	15,098	924

which suggests a duration increase of song flights across breeding season (Table 2).

Data collected in the same study allowed the estimation of the average number of song flights executed per day and thus, it was possible to calculate the frequency in which individuals perform song flights for the average day of each month (Table 2).

While individuals are spread in patches belonging to their territory or outside of these areas for a shorter period than 20 s (*Out_of_territorry* < 20 s), the *Time_count* variable is activated adding 1 unit per second to the counter. *Skylarks* will move randomly on the ground (*Height* = 0) until the *Time_count* reaches the value set for the *Flight_frequency* variable according to the current month, thus beginning the song flight procedure. When individuals remain outside of their territory for a period equal or longer than 20 s (*Out_of_territorry* ≥ 20 s), a comeback procedure is triggered and they will automatically execute a straight movement towards the nearest patch of their territory. In these cases, to prevent song flights outside of the territory, a 10 unit subtraction to the *Time_count* variable is executed, thus ensuring a sufficient time period for the individuals to return to their territory before executing their next song flight. On the other hand, if an individual moves to Class 2 habitat patch (i.e. with unfavourable conditions for territory establishment – area of non-suitable habitat), procedures are triggered involving the automatic return to its territory, based on the assumption that these types of habitats also hold disadvantageous characteristics to perform the song flights, as they are characterized mainly by high vegetation (Donald, 2004).

When the *Time_count* variable reaches the monthly *Flight_frequency* value, several procedures responsible for the song flight are prompted, namely the setting of the variables: *Avoid* (influences the probability of collision related mortality); *Song_flight* (defines the total duration of the song flight) and *Flight_angle* (defines amplitude of the circular trajectory of the flight). The flight song will then take place with two possible pattern movements depending on the wind speed (class 1 – low wind speed or class 2 – strong wind speed). Accordingly, in class 1 wind speed days, individuals will perform the upward and horizontal stages of the song flights with a circular trajectory. Conversely, if the wind speed variable of the simulated day is stronger (class 2), individuals will perform all of the song flight in the same patch, thus covering a smaller area.

According to Hedenström and Alerstam (1996) the average height attained in the vertical ascent flight is 120 m, with a minimum of 30 m and a maximum of 210 m. The authors also found that there is a positive relationship between the maximum height reached and the length of the song flight. The vertical ascent is performed at 1 m/s average speed (Hedenström, 1995). After the upward movement birds will perform a horizontal flight motion with 6 m/s speed which will finally terminate in a descending flight that most commonly occurs by parachuting while singing with an average speed of 1.5 m/s (Hedenström, 1995). In the simulations, the height reached during the aerial exhibition depended on the total length of the song flight (*Song_flight*), previously set for each month. The height reached in the song flights were defined randomly between a minimum of 30 m and a maximum which enabled the execution of both ascending and descending flights, respecting the prearranged flight speeds. Thus, the individuals increased the *Height* variable 1 unit per second which simulated the average ascending speed of 1 m/s described by Hedenström (1995). Subsequently individuals performed a horizontal flight phase, followed by the downward flight with a decrease of 1.5 units per second to the *Height* variable in order to simulate a descending speed of 1.5 m/s as described by the same author.

After each song flight, individuals restarted their *Time_count* variable, repeating the previous procedures cyclically. Additionally, song flight procedures could also be triggered by the interaction between individuals, as a synchronized sequence of song flights performed by neighbouring males, frequently observed in this species (Donald, 2004; Raposeira, 2011).

2.2.1.3.1.4. Mortality. Determining the probability that birds in flight have to avoid colliding with wind turbines is a difficult task considering the complexity of all involved factors. Some authors have conducted studies on bird mortality in wind farms using collision risk models (e.g. Smales and Muir, 2005; Whitfield and Madders, 2006; Furness et al., 2016). In some of these studies, the collision avoidance probabilities were assigned arbitrarily between 95% and 99% (Smales and Muir, 2005), or estimated empirically varying between 98% and 100% (Whitfield and Madders, 2006). On the other hand, Furness et al. (2016) included in their study factors that may determine collision risk, such as behavioural differences between adult males, adult females and fledglings, as well as average flight speed and height converted into an estimate of the number of passes made through the rotor swept area. In the present study, taking into account the complexity associated with collision events, we did not assign a fixed probability for its occurrence, rather defining these events from a set of interactions between individuals and risk patches. Thus, mortality events may only occur when individuals pass through the *patches* that surround the wind turbines. In addition, the *Height* variable (corresponds to the vertical position of individuals) must also be encompassed in the area covered by the wind turbines vertical range of influence. If both conditions are met, the model also includes the possibility of collision avoidance which is defined by comparing the *Avoid* variable of the *skylark* (set randomly between 0 and 99 in the beginning of each song flight) and the *Risk* variable of the position *patch*. An individual will only suffer mortality if the *Risk* variable is superior to the *Avoid* variable. The definition of the *Risk* variable assigned to each patch was conducted regarding the position relative to the wind turbine and a global *Initial_risk* variable, which depends on visibility conditions and was estimated based on studies conducted by Smales and Muir (2005) and Whitfield and Madders (2006). Visibility has proven to have a strong influence on bird collision events, and considering that fog formation is quite common in the study area throughout the year (Profico Ambiente, 2002), we implemented three classes of visibility conditions in order emphasize the increased risk of collision with worsening visibility. All *patches* outside de influence zone of the wind turbine did not present collision risk (*Risk* = 0). Furthermore, we considered the existence of a progressive decrease of the *Risk* variable from the central *patch* of the wind turbine towards the more peripheral *patches* taking into account the coverage area of the entire wind turbine structure. We have supported the Risk variable distribution on Hull and Muir (2010) results that show a gradual increase in mortality/collision probability when approaching the innermost sections of the wind turbines influence area. Thus, we applied progressive divisions to the *Initial_risk* variable as to maintain a relation with the initial value assigned to this variable (Fig. 2).

In case of mortality, the individuals are eliminated from the simulation and territory imprints are removed from the patches (*Free* = 0). These areas will then become available for reoccupation by roaming males.

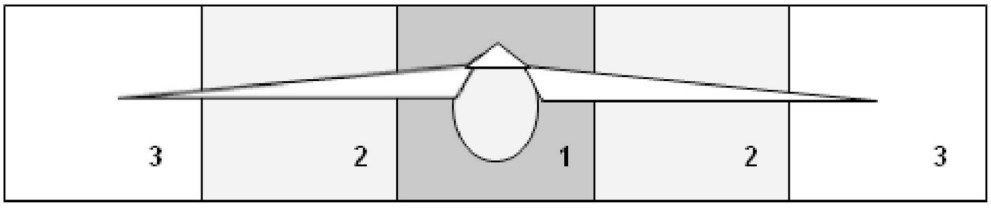

Fig. 2. Distribution of the *Risk* variable for *Alauda arvensis* collision risk represented by the patches in the wind turbine influence zone. 1: *Risk* = *Initial_risk*; 2: *Risk* = *Initial_risk*/2; 3: *Risk* = *Initial_risk*/4.

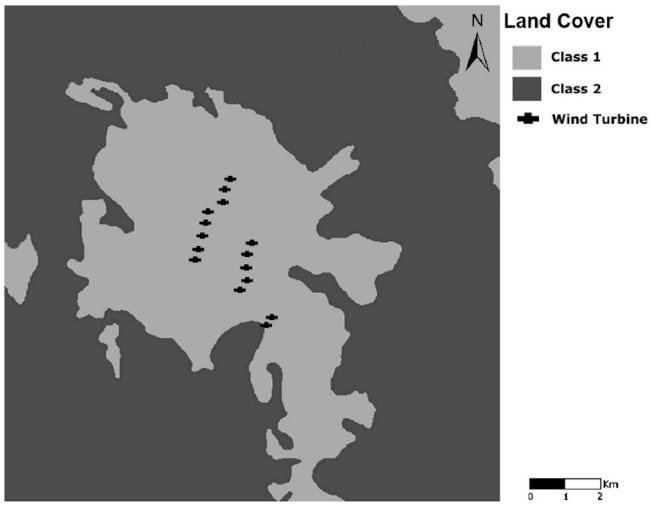

Fig. 3. Representation of the land cover classes encompassed in the study area (Class 1: suitable area for *Alauda arvensis* territory establishment; Class 2: unsuitable area for territory establishment).

2.2.2. Design Concepts

The Design Concepts section of the model description is depicted in Appendix E (Supplementary Material).

2.2.3. Details

2.2.3.1. Initialization. The model reproduces a real landscape located in the north of Portugal: in the beginning of the simulation the raster datasets regarding the land cover and altitude variables are loaded and remain constant during both stages. The variable *Land_cover* (Fig. 3) determines the *patches* land cover and results from the overlap of data from the land cover map for Continental Portugal — Corine Land Cover 2006 (Caetano et al., 2009) and more recent information collected by direct observation of the study site using *Google Earth*®.

The raster dataset for the altitude variable was implemented based on data from the Environment Atlas (http://sniamb.apambiente.pt/webatlas/). All geographic information calculations were performed with the ArcMap 10.0® software. In the beginning of the TES simulations all available areas with favourable altitude and habitat conditions was considered free for territory occupation (*Free* = 0). The data concerning the initial number of active *skylarks*, minimum altitude for territory establishment and allowed territory size range were introduced manually in the beginning of each simulation. In BSS, wind turbine position reflects their position in the real landscape and wind turbine orientation was assigned randomly from 0 to 359°. All wind turbines assumed the same orientation throughout the day.

In the last two decades, the Portuguese wind energy sector has expanded substantially. The total installed capacity by the end of 2016 reached 5258 MW, associated with 2637 onshore wind turbines placed throughout the mainland territory. From 2013 to 2015 a significant portion of 2.0 MW wind turbines were target of an upgrading consisting in the increase of generating capacity to 2.3 MW (APREN, 2016). The comprehensive impact of increase in wind power on wildlife has mainly focused on area avoidance and collision risk (Pearce-Higgins et al., 2012; Loss et al., 2013). Pearce-Higgins et al. (2012) highlighted that repowering of turbines involving an increase in turbine height, might cause little additional negative effects on birds while Loss et al. (2013) found that an increased hub height can lead to higher collision rates. Two different wind turbine models were simulated, namely the 70 m rotation diameter model (rated power 2.300 kW) installed presently in the wind farm, reaching a combined power of 34.500 kW (15 wind turbines): Wind farm structure (WFS) 1. The repowering strategy involves substituting the previous wind turbines by 100 m rotation diameter model (rated power 3.050 kW), projecting a possible generating capacity increase of 11.250 kW (WFS 3). On the other hand, WFS 2 simulates a similar generating power increase resulting from the installation of five additional 70 m rotation diameter wind turbines (equivalent to the actual wind turbines). The criterion for the positioning of the new wind turbines was based on the continuity of the rows already formed by the existing wind turbines along the mountain ridges. Moreover, the minimum functional distances between wind turbines were guaranteed. All the new wind turbines were placed in areas of at least 1100 m, once this criterion was already applied to the real wind turbines of the wind farm, in order to attain maximum wind potential. The *Risk* variable was distributed through the patches in the wind turbines influence area depending on these structures position, size and orientation besides the visibility conditions set randomly for each day. All other patches were considered risk free (*Risk* = 0). Daily wind speed classes were also assigned randomly from the established classes. *Skylark* variables *Song_flight* and *Flight_frequency* were set for each month based on data from Lange (1951). The variable that define the angle of the song flights trajectory as well as the *Avoid* variable were only defined in the beginning of each song flight.

2.2.3.2. TES simulation scenarios. We simulated three minimum altitudes for territory establishment, 800 m, 1000 m and 1200 m, in order to project reductions in the area of suitable habitat (ASH) associated with socio-ecological changes, gauging the number of potential territories for the species. One hundred simulations were run for each ASH tested.

2.2.3.3. BSS simulation scenarios. The initial population of active individuals for the BSS simulations was based on the number of established territories and roaming population of individuals searching for territory (see paragraph 2.2.1.3.1.2. Initial male population configuration). Given that the maximum territory size for this species, in northern Portugal wind farms, does not tend to exceed 5 ha, with an average of circa 4 ha (LEA, 2011; LEA, 2012; Mãe d'Água, 2006), we created the *Buffer* variable that marks a 257 m radius area around the 70 m rotation diameter wind turbines and 272 m radius area around the 100 m rotation diameter wind turbines: wind turbines influence zone. Only individuals with territories encompassed in the buffer area were considered for BSS data analysis.

The BSS simulations were run 10 times for the full extent of the breeding season for each combination of area of suitable habitat (above 800 m (ASH800), 1000 m (ASH1000) and 1200 m (ASH1200)) and wind farm structure (WFS1, WFS2, WFS3). Nine scenarios were considered: WFS1 - fifteen 70 m rotation diameter wind turbines within all ASH; WFS2 - twenty 70 m wind turbines within all ASH; and WFS3 (repowering, i.e. substituting the actual 70 m turbines by 100 m turbines) - fifteen 100 m rotation diameter wind turbines within all ASH.

WFS2 and WFS3 are associated to equivalent increases in wind power.

2.3. Statistics

Since most of the data did not show a normal distribution (Shapiro-Wilk W test), the statistical analysis was conducted using non-parametric tests (Royston, 1993). The Kruskal-Wallis (non-parametric equivalent to the one-way analysis of variance) (Sokal and Rohlf, 1995; Zar, 1999) and the Scheirer-Ray-Hare extension of the Kruskal-Wallis test (non-parametric equivalent to the two-way analysis of variance) (Sokal and Rohlf, 1995; Zar, 1999) was used to verify possible statistical differences in fatalities recorded per total number of established territories (*relative mortality index)*, considering the combined effects of a reduction in the ASH and different WFS. When necessary the Nemenyi multi-comparisons test (Sokal and Rohlf, 1995) was applied to determine, in detail, specific differences. The data analysis for this paper was generated using the Real Statistics Resource Pack software (Release 5.4). Copyright (2013–2018) Charles Zaiontz (www.real-statistics.com).

3. Results

3.1. Territorial selection analysis

The territorial feature analysis was based mainly on the TES outputs. However, territory selection was additionally conducted from data collected in BSS, particularly regarding the area defined by the buffer around the wind turbines: wind turbines influence zone. Skylark territories occupied approximately 79% of the available area of suitable habitat (ASH), regardless of the simulated altitudes (ASH800, ASH1000, ASH1200). The model allowed us to determine the study area's potential for territories within the ASH, which may represent a realistic approximation of the breeding population. The number of territories was calculated for the different ASH, in order to illustrate the impacts of the possible reduction in ASH on the population. The study area has potential to host 583.77 ± 6.22 territories for ASH800 (Fig. 4A), 333.87 ± 5.19 territories for ASH1000 (Figs. 4B) and 47.13 ± 1.36 territories for ASH1200 (Fig. 4C) - the obtained results depict an average density of 0.21 territories per ha for all ASH.

Furthermore, we used the data from the buffer area around the wind turbines (wind turbines influence zone (WTIZ)), attained in the BSS, to determine the average number of territories that this area could host, as we considered that these territories will be the most likely to be affected by mortality events. The reduction in the ASH shifts the population to higher breeding grounds and results in an increasing overlap between the ASH and WTIZ. If we consider the present structure of the wind farm (WFS1), the WTIZ superimposes 7.07% for ASH800, increasing to 12.28% for ASH1000 (with equivalent number of territories within the WTIZ, circa 41 territories) and to 32.36% for ASH1200 (15 territories). WTIZ increases with WFS2, namely 9.52% for ASH800, growing to 16.14% for ASH1000 (56 and 54 territories respectively) and 43.50% for ASH1200 (21 territories). The WTIZ for WFS3 presents values between WFS1 and WFS2, 7.71% for ASH800, 13.48% for ASH1000 (circa 45 territories within WTIZ) and 35.43% for ASH1200 (17 territories).

Table 3
Two way Scheirer-Ray-Hare extension of the Kruskal-Wallis test for the *Alauda arvensis* Relative Mortality Index (RMI) under three areas of suitable habitat (ASH) and three wind farm structures (WFS). Degrees of freedom (df), Sum-of-Squares (SS), Kruskal-Wallis value (H) and associated significance (***, $p < 0.001$; *, $p < 0.05$; n.s., non-significant).

Source of variation	*SS*	df	H
ASH	54,000	2	79.12,479 ***
WFS	4758.217	2	6.97,209 *
ASH x WFS	74.73,333	4	0.109,505 (n.s.)
Error	1906.55	81	
Total	60,739.5	89	

3.2. Mortality analysis

To monitor the impact of the wind farm structure (WFS) on local *A. arvensis* breeding population, the relative mortality index (RMI) was calculated based on the number of fatalities recorded per total number of established territories. For the WFS1, the RMI increases from 8.07% for ASH800 to 13.69% for the ASH1000 and 37.16% for ASH1200. A similar trend was found in the scenarios considering increasing power: i.e. for WFS2 (ASH800: 10.80%; ASH1000: 18.73% and ASH1200: 53.49%) and WFS3 (ASH800: 8.14%; ASH1000: 15.07% and ASH1200: 42.28%). Results show that no significant interaction occurred among the ASH and WFS i.e. these variables do not interact with influence on the RMI outcomes (Table 3).

Additionally, ASH and WFS affect, independently and significantly, the RMI (Table 3). In fact RMI increases as the ASH decreases ($H = 79.121$, df = 2, p-value = 6.58e-06, Kruskal-Wallis nonparametric test (KW); ASH800: ASH1000, $q = 6.29$, p-value = 3.21e-05; ASH800: ASH1200, $q = 12.58$, p-value = 4.00e-14; ASH1000: ASH1200, $q = 6.29$, p-value = 3.21e-05; Nemenyi multiple comparisons test (N)) (Fig. 5).

Although the replacement of smaller turbines with larger devices might result in a slight tendency for higher mortality, the results attained in the BSS simulations showed no statistically significant differences in the RMI considering WFS1 and WFS3. Conversely, statistically differences in RMI were found pointing to higher mortality when more wind turbines were added to the wind farm (WFS1 *versus* WFS2). No statistically differences in the RMI were found between WFS2 and WFS3 ($H = 6.971$, df = 2, p-value = 0.03, KW; WFS1: WFS2, $q = 3.62$, p-value = 0.03; WFS1: WFS3, $q = 1.03$, p-value = 0.75; WFS2: WFS3, $q = 2.59$, p-value = 0.16; N) (Fig. 6).

4. Discussion

4.1. Territorial potential analysis

One of the main objectives of our work was to simulate the complex procedures associated with the individual territory establishment and

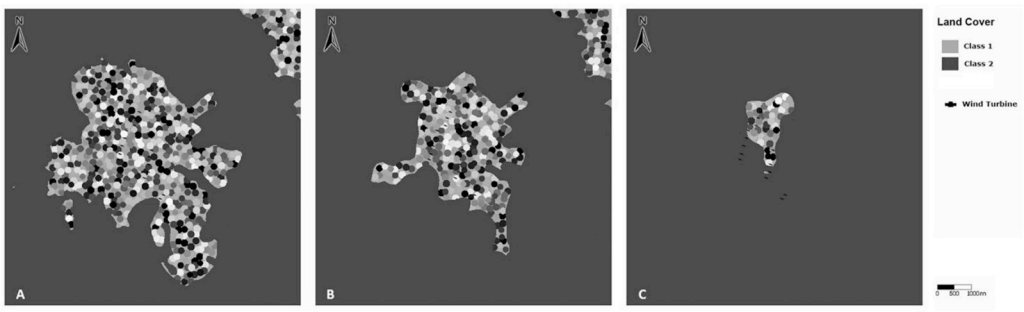

Fig. 4. *Alauda arvensis* territory occupation for the different minimum altitudes considered: Area of suitable Habitat (ASH) A – 800 m (ASH800); B – 1000 m (ASH1000); C - 1200 m (ASH1200). Each different colored area represents a territory (Class 1: area of suitable for territory establishment; Class 2: area of unsuitable for territory establishment).

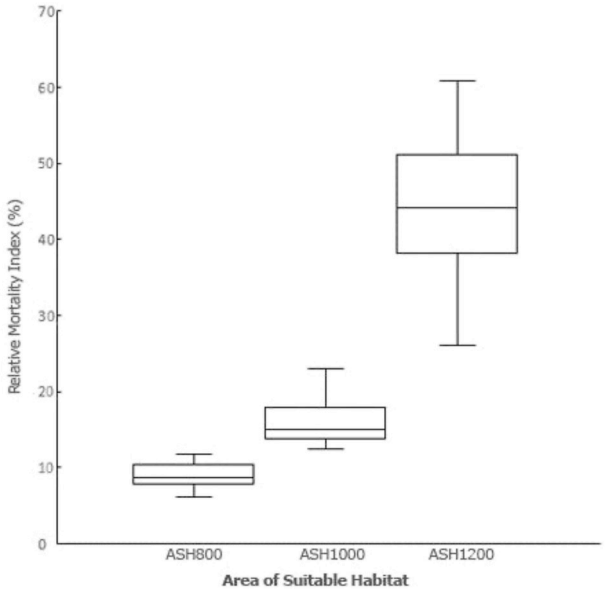

Fig. 5. Box and Whisker plots expressing the differences in the projected values for *Alauda arvensis Relative Mortality Index* (RMI) derived from 90 simulations considering the three areas of suitable habitat (800 m (ASH800), 1000 m (ASH1000), 1200 m, (ASH1200). RMI medians for ASH (ASH800 = 8.71%; ASH1000 = 14.97%; ASH12000 = 44.10%).

defence throughout the breeding season for an *A. arvensis* local population. A realistic performance of model simulation was considered a fundamental requirement for testing the cumulative impacts induced by socio-ecological changes and wind power increase. Different studies point to *A. arvensis* breeding densities ranging from 0.11 to 0.90 territories per ha (Donald, 2004). The density estimates obtained by our model, circa 0.21 territories per ha, were additionally corroborated by works performed in mountain ridges of the same region (Mãe d'Água, 2006; Raposeira, 2011). Several studies have linked the predictable effects of global environmental changes on species distributions, habitat shifts and biodiversity declines (e.g. Araújo et al., 2006; Garcia-Ruiz et al., 2011; Bastos et al., 2016). Anyway, global and local stressors interaction (e.g. climate induced changes and abandonment derived land cover changes) are of extreme relevance when assessing local effects (Jetz et al., 2007; de Chazal and Rounsevell, 2009; Reino et al., 2018), namely for species that have been showing shifts in their altitudinal distribution attributable of global warming (Crick, 2004). The expected upwards displacement of several bird species (García-Romero et al., 2010; Ruiz-Labourdette et al., 2012; Reif and Flousek, 2012; Lehikoinen et al., 2018; Rumpf et al., 2018) as well as the habitats associated (Pauli et al., 2007; Harsch et al., 2009) resulting from novel climates in the Mediterranean region, such as in our case study, cannot be easily disentangled from local habitat dynamics (Giorgi and Lionello, 2008; Lehikoinen et al., 2018). An expected consequence of climate change in mountain regions of the Mediterranean is related with a decrease in precipitation and increase in temperature (Giorgi and Lionello, 2008). This novel climate interacts with grazing regimes, possibly creating less resilient pastures (Rota et al., 2017), promoting its abandonment and consequent substitution by xerophytic shrublands. Nevertheless, the upward shift in xerophytic shrublands is also stimulated by the traditional animal husbandry practices abandonment caused by the depopulation of inland hill and mountain Mediterranean areas (Serra et al., 2008). *A. arvensis* and many other co-occurring species (*e.g. Prunella modularis*, *Emberiza cia*, *Saxicola rubicola*, *Sylvia undata*, *Monticola saxatilis, Anthus campestris* and *Circus pygargus)* are particularly exposed to the mentioned socio-ecological changes, namely in their breeding habitats in the Mediterranean mountains, having to establish their territories at higher altitudes accompanying the uphill shift in the area of suitable habitat (Theurillat and Guisan, 2001; Bastos et al., 2016).

4.2. Mortality analysis

The hypothetical shift from 800 m to 1000 m and 1200 m thresholds in the area of suitable habitat (ASH) resulted in a substantial increase in the proportion of breeding grounds superimposed to the wind turbines influence zone (WTIZ). These results have implications in the relative mortality index (RMI) that raises almost fivefold (8%–37%) considering the current wind farm structure (WFS1). The conversion of RMI results (WFS1 within ASH800) in mortality by breeding season (47.1 ± 3.92 fatalities) depicts comparable results to an ecological impact assessment performed in the same region for the species (49.5 fatalities) (Profico Ambiente, 2007): we strongly believe that these results provide a solid basis for the projected scenarios. Although wind farm impacts already represent a current cause for concern (Masden et al., 2010), their persistent infrastructural impacts combined with species distribution displacements may further influence population dynamics (Morinha et al., 2014; Bastos et al., 2016), thus highlighting the importance of long-term ecological assessments (Roscioni et al., 2013; Santos et al., 2013; Bastos et al., 2016). The significant worldwide investment in the wind energy sector (Silva et al., 2006; Forte et al., 2010; Ferreira et al., 2015; Santos et al., 2017) was also clearly noticed in Portugal, with a strong tendency for adding new turbines within each wind farm for increasing generating capacity (APREN, 2016). We evaluated the effects of this projected growth by testing two possibilities, either replacing the wind turbine models present in the wind farm by larger rotation diameter models (repowering) or adding more wind turbines to attain the equivalent wind power. Our results showed similar mortality rates for both wind turbine models. Anyway, considering the fewer turbines involved, repowering may represent a weaker aggravation on the pre-existing mortality caused on a population of *A. arvensis*, even regarding the possible impacts caused by a reduction of the ASH due to socio-ecological changes. This tendency has also been found in several other studies (e.g. Pearce-Higgins et al., 2012; Hötker et al., 2006; Stewart et al., 2007; Thaxter et al., 2017). Nevertheless, specific wind turbine features such as increases in hub height have been correlated to higher collision rates (Loss et al., 2013). In-depth investigation on this correlation is even more relevant as wind turbine devises up-scale (Pearce-Higgins et al., 2012; Bastos et al., 2016). On the other hand, installing additional wind turbines increases the wind farm influence zone, with direct implications on collision probability for flying vertebrates, particularly relevant considering the current threats to wildlife in mountain areas.

4.3. Implications for conservation of vulnerable mountain habitats and species

Several studies highlight species global response to climate change by shifting geographic distributions (Parmesan and Yohe, 2003; Chen et al., 2011; Elsen and Tingley, 2015). Although shifts may occur in a heterogeneous way (Tingley et al., 2012), upslope pattern displacements have been most commonly attributed to warming climates (Parmesan and Yohe, 2003; La Sorte and Jetz, 2010; Elsen and Tingley, 2015). Climate induced range shifts might be enhanced when local anthropogenic actions create rapid habitat modifications associated with abandonment or intensification of land uses (Rumpf et al., 2018). The interplay originates a progressive reduction in the area of suitable habitat, affecting particularly mountain species as they may be facing increasing risk of population isolation and local extinction (Jetz et al., 2007; Colwell et al., 2008; La Sorte and Jetz, 2010; Sekercioglu et al., 2012; Triviño et al., 2013; Flousek et al., 2015). The model proposed for the species breeding behaviour might be considered as a step forward in anticipating the local consequences of wind farms on *A. arvensis* populations in order to test different alternative strategies facing the

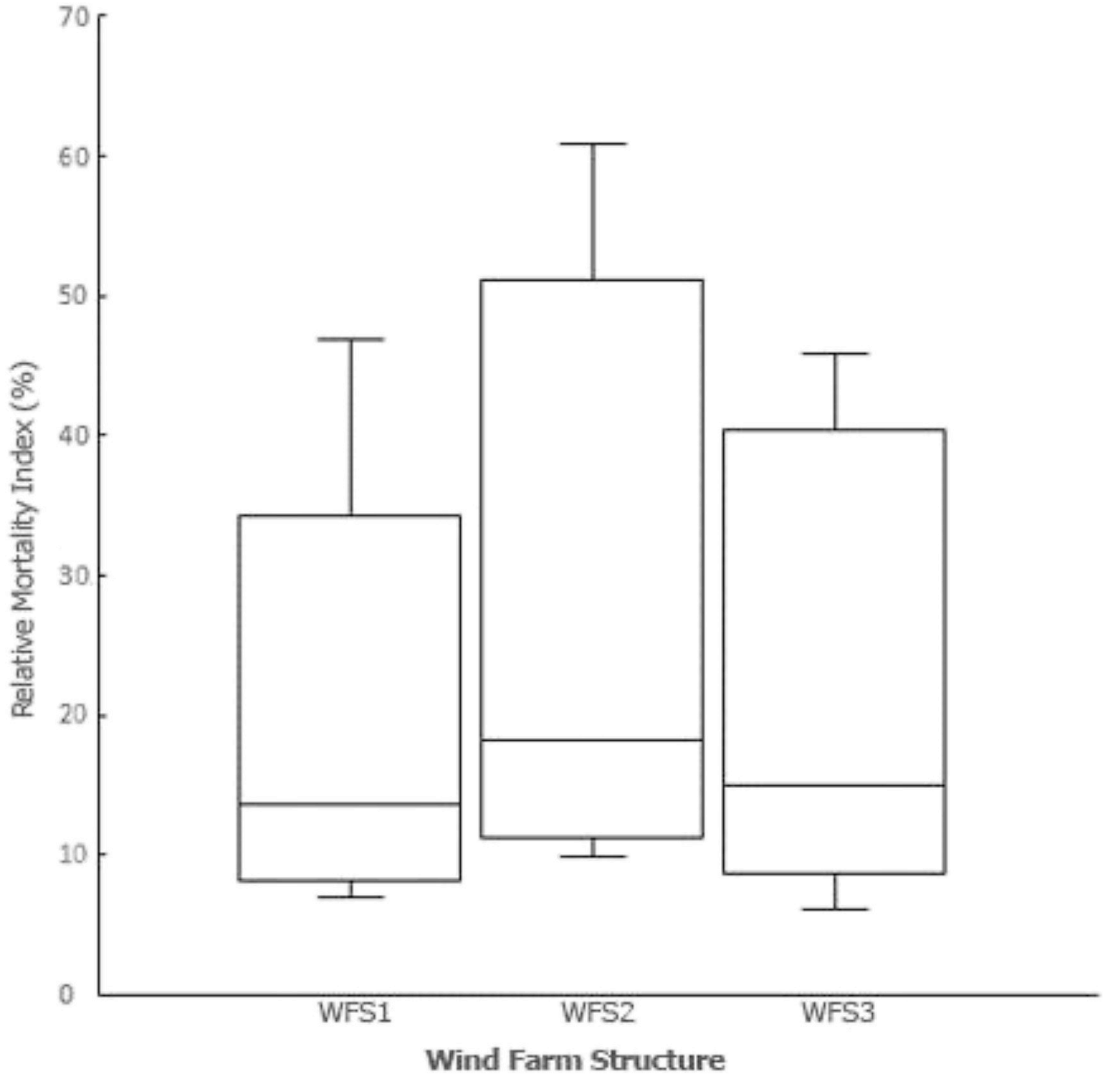

Fig. 6. Box and Whisker plots expressing the differences in the projected values for *Alauda arvensis Relative Mortality Index* (RMI) derived from 90 simulations considering the three wind farm structures (WFS). RMI medians for WFS (WFS1 = 13.60%; WFS2 = 18.17%; WFS3 = 14.97%).

increasing wind power while mitigating the respective ecological impacts, namely by: a) assessing direct impacts from the installation of new wind turbines/repowering in pre-existing wind farms; b) assessing indirect impacts, such as habitat loss and environmental changes, as demonstrated by the reduction in the number of territories associated with shifts in the area of habitat suitability; c) assessing the cumulative impacts of both direct and indirect impacts in the trends of vulnerable declining species such as the case of *A. arvensis*.

5. Conclusions

The modelling approach developed presents unique features that enable understanding the processes associated with declines in an *A. arvensis* population, resulting from individual responses to both global environmental changes and local anthropogenic stressors. The methodology was considered useful in the scope of strategic options for wind power developments: the outputs recreate with realism the territorial establishment process of *A. arvensis* within evolving landscapes, the territorial behaviour during the breeding season and its implication on the mortality risk associated with wind farm characteristics. Nonetheless, significant parameterization and validation using field data is vital for using ABMs in concrete planning and design processes, namely in the scope of wind power impact assessments. We also envision that including ABMs within frameworks with traditional modelling approaches (e.g. process-based models and/or species distribution models) will result in promising future outcomes with increasing predictive power, more instructive and credible to decision-makers and environmental managers. The principles of the approach proposed may be applicable to other case studies within certain limits and after the necessary adaptations to the base code.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2019.02.127.

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